Discussion paper: Policy on food web modeling at hazardous waste sites

The use of simple food web models is a popular but largely untested method for estimating risks to terrestrial wildlife at hazardous waste sites[1]. In ecological modeling, as in other disciplines, testing of the model with empirical data (model validation) is an important prerequisite for demonstrating its scientific validity[2]. This is generally not practical for hazardous waste site investigations, however, and the problem is further compounded both by the limited amount of relevant information in the literature, and uncertainties regarding its applicability to particular current and future conditions at a specific site. With broad latitude on how to proceed, food web models can be used to arrive at a wide range of results that could potentially translate into risk-based cleanup levels that vary considerably.

Despite these weaknesses, there is a need for relatively inexpensive, short-duration methods for reaching timely remedial decisions that are protective of the ecological environment. The following discussion explores the problem of how food web modeling, as practised in hazardous waste site investigations, can be adapted for regulatory decision-making despite its scientific shortcomings.

I. Evaluating risk assessment methodologies.

There are two key criteria for evaluating the usefulness of a particular risk assessment methodology in conducting meaningful ecological risk assessments:

1) <u>Does the methodology produce repeatable results?</u> Given the same information about a site, would different risk assessors with comparable expertise obtain similar estimates of risk? There is low confidence in a methodology that yields widely varying estimates of risk for a particular site, depending on the assumptions and policy decisions used by different risk assessors. For food web modeling, these include decisions on which species to include in the model and what values to use for each parameter in the model.

2) <u>Do differences in *estimated* risk at different sites reflect *actual* differences in risk?</u> Actual ecological risks at different sites can vary considerably due, for example, to variation in the toxicity or bioavailablity of soil contaminants, and risk assessments should reflect such differences. However, where the differences in estimated risk are largely or entirely due to differences in assumptions and policy decisions made by different risk assessors, there is low confidence in the methodology. Results from food web modeling are typically very sensitive to underlying assumptions and policy decisions.

II. Conclusions

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[1] Some exceptions include testing of specific applications by Hendriks and others in The Netherlands (e.g., Ma and van der Voet, 1993; Hendriks et al., 1995) and Pascoe et al. (1994).

[2] As discussed by Suter (1993), "validation" means establishing how well the model corresponds to reality, not whether the model corresponds exactly to reality.

A food web modeling methodology can be the basis for credible ecological risk assessments if the following conditions are met:
1) The methodology yields consistent results when applied to the same site by different risk assessors with the same information.
2) Differences in estimated risk at different sites largely convey information about the site, rather than information about the risk assessor's assumptions and policy decisions.

In principle, both of these conditions can be met if the results from food web modeling are dependent on objective, empirical site-specific data. The conditions are not met if the results are largely dependent on underlying assumptions and decisions that can vary, depending on the risk assessor performing the evaluation.

III Recommendations

The following recommendations are provided to meet the conditions described above.

A. What components of the food web model methodology should be held constant?

To meet the conditions listed under section II, components of the model that involve assumptions and policy decisions should be consistently applied at every site. These components generally fall into two areas: decisions relating to uncertainty and the selection of receptor species.

1. Uncertainty. Where there are uncertainties in the modeling, a consistent approach is needed (e.g., on what value should be used for a particular parameter), since the modeling results are likely to be very sensitive to these decisions.

Food web modeling entails many sources of uncertainty[3]. Some arise, first, from the model itself. Typical food web models are highly simplified representations of complex and dynamic biological systems. Although they can provide a rough approximation of risk, they cannot make detailed predictions. Temporal changes in risk (e.g., Hunter et al., 1987) are not considered, for example. While all of the available models suffer from these limitations, the choice of which model to use is one factor that can affect the outcome of the risk assessment. Although more complex and site-specific models could be developed, the time and expense required to support use of the model would be prohibitive for a MTCA Remedial Investigation.

Parameter values. The selection of a value for each parameter (variable) in the model requires a decision that will affect the results from the modeling. The uncertainties involved are described below.

The sensitivity of most wildlife species to toxic chemicals is unknown. Toxicity Reference Values (benchmarks) are typically only available for a few species, and extrapolating this

[3] Suter (1993) describes the three types discussed here as natural stochasity, parameter error and model error.

information to a species of interest involves a high degree of uncertainty. Other related areas of uncertainty include age-dependent differences in sensitivity, the toxicity of multiple toxic chemicals, and the bioavailability of toxic chemicals in ingested food and soil.

There is high uncertainty concerning biologically appropriate values for exposure parameters in the model. In particular, the uncertainties relate to the following issues:

- What proportion of the diet is contaminated by toxic chemicals?
- What levels of contamination are present in the food?
- How much soil is ingested?
- What is the duration of exposure?

Natural variablity accounts for additional uncertainities. For example, data on diet composition from one locality may not be representative of feeding activities at another location. Seasonal and age-dependent differences in exposure may also vary in ways that are poorly understood and not accounted for in the model.

2. Receptors. The choice of species to be included in the food web model is another factor which can significantly affect results from a risk assessment based on food web modeling. For example, standards for most, if not all, commonly encountered soil contaminants that are based on the protection of herbivorous mammals are unlikely to be protective of predators. This generalization is based both on a survey of risk-based standards established using food web models (e.g., Efroymson et al., 1996) and empirical studies (e.g., Ma et al., 1991). Other generalizations from this survey are:

- Standards based on the protection of large mammals may not be protective of small mammals. In the (usual) absence of species-specific toxicity data, allometric scaling considerations predict lower sensitivity to toxic chemicals in small-bodied mammals. However, overall risk is higher due to higher exposure associated with small size (e.g., due to smaller home range and higher ingestion rates per unit body mass).
- Standards based on the protection of mammals may not be protective of birds. Toxicity Reference Values (TRVs) for birds may be lower than for mammals (see Table 1).

Table 1. Comparison of mammalian and avian TRVs (mg/kg - d) obtained from the literature in establishing Tier II soil screening levels for the protection of wildlife.

Chemical	Mammalian TRV (Insectivore)	Avian TRV (Insectivorous bird)	More sensitive receptor
Aldrin	2.2	0.06	B
Arsenic III	1.9	22	M
Arsenic V	35	22	
Benzene hexachloride	28.1	2.3	B
Cadmium	15	20	
Chlordane	10.9	10.7	
Copper	44	61.7	
DDT	8.8	0.87	B
Dieldrin	0.44	4.37	M
Dioxin	2.2e-5	1.4e-4	B
Lead	20	11.3	
Mercury - inorganic	24	9	B
Mercury - organic	3	0.6	B
Nickel	175.8	107	
PCBs	2	5	
Selenium	0.8	4.3	M
Zinc	2,445	570	B

Where there are large (order of magnitude) differences in avian and mammalian TRVs, the more sensitive receptor is indicated in the last column (B bird, M mammal).

Because of the sensitivity of food web modeling results to the choice of receptor species, Ecology has standardized the receptors to be included. The following groups (guilds) should be represented:

1) Small mammalian predator whose diet includes invertebrates living in close contact with soil.
2) Small mammalian herbivore.
3) Ground feeding avian predator whose diet includes invertebrates living in close contact with soil.

The choice of these receptor groups is based on the following considerations:

1) Standards based on protection of these groups are likely to be protective of wildlife in general. Standards based on protection of receptors such as large mammalian herbivores may not be protective of other wildlife species, for the reasons discussed above.
2) The three receptor groups are typically represented in terrestrial ecosystems. Where there is good evidence that any of these groups is not part of the regional ecology (at the landscape or ecosystem level), it need not be included.
3) Where there a site-specific concerns regarding risks to a particular species, the receptor may be added to the food web model.

Surrogate species. To insure consistency in policy components of food web modeling, standard exposure assumptions have been developed for surrogate species for the three receptor groups. The surrogates are listed below:

Group 1. *Sorex* spp.
Group 2. *Microtus* spp.

Group 3 American Robin (*Turdus migratorius*)

For Groups 1 and 2, standard exposure assumptions are based on the biology of *Sorex vagrans* and *Microtus pennsylvanicus*, respectively. The biology of these species is expected to be representative of that of their respective congeners.

The primary criteria in selecting surrogate species are (1) whether adequate information is available in the literature on the biology of the species, and (2) a preference for surrogates that are widely distributed in Washington state.

For Groups 1 and 3, earthworms are used as surrogates for soil invertebrate prey in estimating contaminant concentrations in food. There is a relatively large body of literature on the bioaccumulation of toxic chemicals by earthworms. Information for other soil invertebrates is scant.

Substitution of surrogate species.

As a risk assessment methodology, food web modeling has low resolution for discriminating between risks to ecologically similar species (i.e., members of the same Group or guild). Due to the limitations of the method discussed above, estimates of risk for any of the surrogate species for the three Groups described above have an associated uncertainty that may be larger than differences in risk among other members of the same Group.

Where substitutions are made for the selected surrogates, it will be difficult to establish whether resulting differences in estimated risk reflect biological differences or are simply due to policy differences relating to uncertainties. Does a lower risk estimate for a substitute species reflect biological differences or the use of less conservative assumptions in the choice of food web values for the substitute?

The preceding are difficult issues that should be considered by both the proponent for a substitution and by the regulatory reviewer. Because of the inherent sensitivity of food web modeling to policy assumptions and decisions relating to uncertainty, the substition of an ecologically similar species can lead to significant changes in the outcome of the risk assessment when accompanied by the use of less conservative assumptions for the substitute species. Confidence in the predictive value of models normally rests on testing conducting during model development (model validation). The model validation step is rarely conducted, however.

Food web modeling includes large elements of policy and science. Because the policy element relates to level of environmental protection, it falls within the responsibilities of the regulatory agency, rather than the risk assessor. Elements of the approach that rely on objective empirical measurements that are largely science can form the basis for demonstrating significant site-specific differences in risk from the outcome using default values in the standard model. Measurements of site-specific toxicity and bioavailabilty of soil contaminants are examples. The latter is particularly amenable to site-specific study, by measuring contaminant concentrations in prey species.

B. What components of the food web model methodology should be subject to change?

The following are suggested criteria that should all be met to satisfy the conditions described in Section II:

1. The modification is based on site-specific information.
2. The modification is based on objective measurements. In practical terms, this means that different risk assessors should be able to reproduce the proposed basis for a change to the model. A site-specific, empirical estimate of contaminant bioavailability is likely to be repoducible by different risk assessors, for example.
3. The proposed modification reflects characteristics of the site that are unlikely to change over time. Characteristics of the soil contamination such as bioavailability could meet this criterion, for example. On the other hand, receptor characteristics may not be stable over time. These characteristics include the species present at the site - a common species not currently present may occur there in the future, for example - and dynamic characteristics, such as population size, home range and feeding behavior.

Arid land ecology

The food web structure of steppe region communities in Washington may be sufficiently distinct that the default food web model does not adequately characterize risks to wildlife, even when the range of uncertainty is taken into account. Ecology is currently exploring the possibility of developing a food web model in collaboration with the U.S. Department of Energy for use at Hanford that could also be used at MTCA sites located on arid areas within the steppe regions of Washington.

References:

Efroymson, R.A., G.W. Suter, B.E. Sample and D.S. Jones. 1996. Preliminary Remediation Goals for Ecological Endpoints. U.S. Department of Energy. ES/ER/TM-162/R1. (Available at http://www.hsrd.ornl.gov/ecorisk/ecorisk.html). To illustrate the higher predicted risks for predators versus herbivorous mammals, compare, for example, the PRGs for the white-footed mouse and the short-tailed shrew (Table 6), which are similar in body size.

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